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Model-based analysis of nutrient retention and management for a lowland river

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Abstract

In the context of the European Water Framework Directive options for improving the water quality of the lowland river Havel (Germany) were assessed. The lower section of this river is actually a polytrophic river-lake system suffering from high external nutrient loading and exhibiting significant in-river turnover. In order to gain a better understanding of present conditions and to allow integrated scenarios of nutrient management to be evaluated the catchment models SWIM and ArcEGMO-Urban were coupled with a simple, newly developed nutrient TRAnsport Model (TraM). Using the TraM model, the retention of nitrogen and phosphorus in a 55 km reach of the Lower Havel River was quantified and its temporal variation was analyzed. It was examined that about 30% of the external nitrogen input to the Lower Havel is retained within the surveyed river section. A comparison of simulation results generated with and without consideration of phosphorus retention/release revealed that summer TP concentrations are currently increased by 100–200% due to internal loading. Net phosphorus release rates of about 20 mg P m^{-2} d^{-1} in late summer were estimated for the Havel lakes. Scenario simulations with lowered external nutrient inputs revealed that persistent phosphorus limitation of primary production cannot be established within the next decade. It was shown that a further reduction in nitrogen concentrations requires emissions to be reduced in all inflows. Though the TraM model needs further extension it proved to be appropriate for conducting integrated catchment and river modeling.

1. Introduction

In many river basins eutrophication appears to be one of the most important water quality problems. In search of adequate restoration strategies eco-hydrological models are frequently used to assess the impact of landuse management on nutrient emissions from river basins (e.g. Schreiber et al., 2005; Kronvang et al., 1999). However, the representation of individual water bodies in catchment models is often poor. This is especially true for regulated rivers and river-lake systems, which exhibit a unique behavior with respect to nutrient retention. Since the European Water Framework Directive (WFD) focuses on the ecological status of individual river sections and lakes, there is an increasing need for linking catchment models to water quality models of adequate space-time resolution and complexity (Van Griensven and Bauwens, 2003).

Within a study on the water quality of the Havel River (NE-Germany) a conceptual nutrient TRAnsport Model (TraM) adapted to lowland rivers was developed. It provides a suitable method to integrate the output of catchment models into a simple transport scheme and enables in-river concentrations of total phosphorus (TP) and total nitrogen (TN) to be estimated. The model was applied to the Havel River downstream of Berlin (Fig. 1) with two goals in mind: First, the relevance and seasonality of nutrient retention had to be figured out quantitatively by evaluating data from official water quality monitoring programs. Secondly, the impact of different strategies of catchment management on total nutrient concentrations of the river was assessed.

The studied river section is characterized by a large number of interconnected lakes and is regulated by weirs. Since external nutrient input is significant, the Lower Havel River is strongly eutrophic and in-river turnover of nitrogen and phosphorus was expected to be extraordinarily high. Although nutrient loading decreased in the 1990s, phosphorus concentrations remained high, because a large P-pool has accumulated in the lakes' sediment over the last decades. Today the lakes act as net sources of P as it is typical after reduction of nutrient loads in the inflow (Søndergaard et al., 2003; Kozerski and Kleeberg, 1998; Jeppesen et al., 1991).

This paper introduces the basic concept of the nutrient transport model TraM and its application within a catchment modeling framework. A special focus is put on the estimation of N and P retention with high temporal resolution and its representation in the model. Selected results of scenario analyses carried out with TraM are presented and discussed in the context of water quality management.

2. Methods

2.1. The transport model TraM

In the TraM model water bodies can be represented by two basic concepts commonly used in water quality modeling (Chapra, 1997): River reaches where advective transport is dominant are represented by plug-flow reactors (PFR) whereas the shallow well mixed lakes shown in Fig. 1 are approximated by continuous flow stirred tank reactors (CFSTR). Table 1 summarizes the features of both types of reactors as implemented in the TraM model.

The first step in setting up the model is to subdivide the river system into a number of PFR and CFSTR. Longer river reaches are further discretized into separate PFR since every reactor is parametrized by a single average cross-section. As the water surface slope of impounded lowland rivers is usually no more than a few cm/km a single stage hydrograph is assigned to every PFR. The length of a plug flow reactor usually falls in the range from less than 100 to several hundred meters. The system shown in Fig. 1 was split into 88 reactors. To form a network with defined upstream-downstream relations suitable for routing calculations, all CFSTR and PFR are linked automatically based on GIS data.

At runtime, the load hydrograph at the downstream end of a reactor is computed using the static and dynamic input data listed in Table 1. Within a complete model run, the computation is carried out for each single reactor and advances from the upstream end(s) of the river network to the downstream model boundary. In the current version of TraM only a single substance can be simulated at a time and retention is described according to zero or first order kinetics (see Sect. 2.3).

2.2. Linking river transport, hydrodynamic and catchment models

As depicted in the lower part of Table 1 hydrological time series as well as data on river loads are needed to run a TraM simulation. For model calibration observed discharge and concentration hydrographs were provided by the Brandenburg state water authorities. However, in case of scenario analyses all boundary conditions must be simulated as well. In this study, nutrient loads and discharges that enter the system shown in Fig. 1 were calculated by a catchment modeling work group using the models SWIM (Krysanova et al., 2000) and ArcEGMO-Urban (Biegel et al., 2004). SWIM computes the discharge of tributaries and the corresponding P and N loads resulting from non-point emissions. In contrast, the Urban module of ArcEGMO estimates N and P losses from point sources only. Discharge and nutrient loads of the tributary Spree were approximated based on observed flow and concentration data also for scenario simulations. The internal flow distribution as well as stage hydrographs for the investigated river section were simulated with the unsteady 1D hydrodynamic model HEC-RAS (USACE, 2002). It proved to be particularly suitable because it handles looped river networks and supports the input of geometric data via GIS. The HEC-RAS data base for the system shown in Fig. 1 comprises about 1100 cross-sections and 27 junctions. The interaction between the applied models is illustrated in Fig. 2.

2.3. Estimation of nutrient retention

Water quality is monitored biweekly by the authorities at many stations along the Lower Havel River (Fig. 1). To assess nutrient retention, data from the period 1991–2004 were used. Since this study focused on total nitrogen and total phosphorus no differentiation into distinct N or P species was made. Nitrogen losses are mainly caused by denitrification and sedimentation (Jensen et al., 1992b; Seitzinger, 1988). Both, nitrate reduction (Chapra, 1997) and settling of particulate N (Scheffer, 1998) are dependent on the respective concentrations. Since Jensen et al. (1992b) found denitrification in lakes to be closer related to TN concentrations (C_{TN}) rather than to NO_3^-, a lumped first order term was tested for the description of total nitrogen retention (Eq. 1).

$$-\frac{dC_{\mathrm{TN}}}{dt} = k_{\mathrm{TN}} * C_{\mathrm{TN}} \quad (1)$$

Initial estimates of the loss rate k_{TN} (d^{-1}) for all major lakes were deduced from Eq. (2).

$$k_{TN} = \frac{Q}{V} * \frac{(C_{TN,obs} - C_{TN,sim})}{C_{TN,sim}} \tag{2}$$

$C_{TN,obs}$ is the observed concentration ($mg\ l^{-1}$) at the lake outlet and $C_{TN,sim}$ is the corresponding concentration that can be calculated from the TN load in the inflow treating the lake as a conservative CFSTR. Q is the flow rate ($m^3\ d^{-1}$) and V (m^3) is the lake volume. For the river reach between the monitoring stations 1 and 2 (Fig. 1) k_{TN} could be estimated from the ratio of total nitrogen concentration (C_{TN}) at these stations using the corresponding mean travel time T_m (d). The calculation is based on Eq. (3) which can easily be derived from Eq. (1).

$$k_{TN} = \frac{1}{T_m} * ln\left(\frac{C_{TN,upstream}}{C_{TN,downstream}}\right) \tag{3}$$

T_m was determined for a range of steady flows by simulating the propagation of artificial load pulses. Monthly values of k_{TN} were estimated according to Eq. (3) using MQ_{month} in the calculation of the mean travel time T_m. Monthly median values of the concentration ratio were used to account for outliers in the TN data. The estimates of k_{TN} obtained by the above methods were further refined in manual model calibration. The objective was to achieve best agreement of observed and simulated monthly median TN concentrations at 12 monitoring stations shown in Fig. 1. One set of monthly k_{TN} values was calibrated for each lake or river reach between neighboring monitoring stations, respectively.

The seasonal dynamics of phosphorus retention r_{TP} was obtained from mass balances for the five largest lakes over the years 1995–2000. r_{TP} was calculated similarly to k_{TN} by relating the observed phosphorus load at the lake outlet ($L_{TP,obs}$) to values that could be expected due to the P load in the lake's inflow ($L_{TP,sim}$), assuming the lake to be a conservative CFSTR. The difference ($L_{TP,obs}-L_{TP,sim}$) was divided by the lake's surface area to yield values of r_{TP} in (mg P m^{-2} d^{-1}). Negative values of r_{TP} indicate P retention whereas in times of net phosphorus release r_{TP} becomes greater than zero. Phosphorus retention in river and channel sections was neglected in this study, because the sediment surface area is small and residence time is short compared to that of the lakes. Also, it is known from on site inspection that P enriched organic sediments rarely exist in the river and channel sections.

3. Results and discussion

3.1. Significance of phosphorus retention and release

The seasonal dynamics of phosphorus retention as it was calculated from mass balances (see Sect. 2.3) is shown in Fig. 3. A phenomenon observed in all investigated lakes is a period of net phosphorus export from July–November with maximum net release rates from August–October. In winter and spring the settling of particulate phosphorus balances the phosphorus release from the sediment or even results in slight net retention. Analogous seasonal patterns were identified for a river-lake system in the nearby Nieplitz catchment. The net release rates shown in Fig. 3 result in a massive increase of pelagic P concentrations during summer. Phosphorus limitation of primary production does not occur under these conditions.

Reliable predictions of future phosphorus concentrations require the lakes' sediments to be included in TraM as an additional model compartment. This is necessary as in the long term the P pool in the sediment will decrease due to continued net phosphorus export (Fig. 3) and thus P release will do. Although corresponding models have been developed in the past (e.g. Van der Molen, 1991) the establishment of transferable relations between sediment characteristics (including P content) and internal loading is still a major challenge of limnological research (Søndergaard et al., 2003). The development, parametrization and calibration of a sediment P model for the Havel lakes was beyond the scope of this study.

However, since scenario simulations for the time period 2003–2015 were requested, reasonable assumptions on future P release had to be made. Therefore, annual net phosphorus export rates from the Havel lakes were related to the amount of inorganic bound P stored in the upper 30 cm sediment layer. This layer seems to be relevant for P export, because upward diffusion of phosphate in the sediment pore water reaches down to a maximum depth of about 30 cm according to SRP profiles (Schettler, 1995). The sediment P and Fe contents were estimated from core samples collected in summer 2004. Balance calculations revealed that several decades of continued net P export will be necessary to cause a decrease in the sediment P/Fe ratio to a value of about 0.12 (atomic ratio). The latter ratio is of special relevance since iron hydroxides very effectively adsorb phophorus especially under oxic conditions, making the P/Fe ratio an indicator for P release potential of the sediment. Jensen et al. (1992a) and Maassen et al. (2005) found that sediments with P/Fe ratios below 0.16–0.12 show significantly decreased phosphorus remobilisation. In addition one has to consider that in connected river-lake systems the phosphorus exported from one location will partly settle in lakes further downstream causing a delay in "self-purification". Consequently, all scenario simulations discussed in Sect. 3.4 were carried out on the assumption that phosphorus release rates shown in Fig. 3 remain unchanged until 2015 (see also Sect. 3.3).

In Fig. 4 simulated concentrations of TP at the monitoring station Ketzin (label 1 in Fig. 1) are plotted against biweekly observations. As expected, the goodness of the model prediction (solid graph in Fig. 4) varies from year to year due to the use of average P retention rates (see Table 3). Actually, the seasonal succession of plankton shows significant variability and certain groups of algae make different proportions of phytoplankton biomass every year. E.g. one could expect net P release rates above average when algae with low settling losses become dominant because less particulate P is retained. This mechanism might be an explanation for the heavy underestimation of P concentrations by the model in summer 2000 (see Fig. 4) as the biomass of cyanobacteria was extraordinarily high in this year according to official monitoring data. However, the effect of sedimentation on net P export is ambiguous, since a high settling rate may increase P retention on the one hand but it may favor the release of redox-sensitive bound P on the other hand by stimulating mineralization. In addition, short periods of temporary stratification may occur in some years. These events can hardly be predicted but may be of relevance for the redox conditions at the water-sediment interface and thus for P release (Welch and Cooke, 2005; Kleeberg and Kozerski, 1997).

Against this background, it seems too early to speculate on the strong overestimation of P concentrations in summer 2003 and 2004 by the model (Fig. 4). The next years will prove, if this can be attributed to interannual variability or whether the thesis (see above), according to which the effects of todays net P export will not become perceptible before decades, needs correction.

As can been seen in Fig. 4 the conservative model which neglects phosphorus retention (dashed graph) and the calibrated model produce similar results in winter and spring, indicating that P concentrations are almost completely attributed to external loads during these seasons. But from July onwards, the model results heavily diverge and it can be stated that internal loading increases summer TP concentrations by a factor of 2–3 compared to a situation without net phosphorus release from sediments.

3.2. Significance of Nitrogen Retention

Since the underlying processes of nitrogen losses were not differentiated, the calibrated values of the retention parameter k_{TN} (Eq. 1) reflect the effects of both denitrification and N sedimentation. As shown in Fig. 5 nitrogen retention is low in winter with k_{TN} in the range 0–0.02 (d^{-1}). In summer N losses range from 0.005–0.03 (d^{-1}) in most sections of the river-lake network. Maximum values of about 0.04 (d^{-1}) indicate that up to 4% of total nitrogen may be eliminated in one day. The absolute values of N retention (Table 2) are of the same magnitude as those reported from shallow Danish lakes (Windolf et al., 1996).

Figure 5 shows conspicuous differences in magnitude and seasonality of TN retention between water bodies in terms of k_{TN}. A satisfying explanation of the observed patterns would require more detailed investigations that do not focus on total nitrogen alone but include all relevant N species. However, some first hypothesis shall be given here:

One obvious feature emerging in lakes B, C, E, and F is that highest nitrogen losses occur already in spring, long before water temperature or plankton biomass reach their maximum values. Possibly, this could be explained by the dominance of diatoms in spring and early summer. Due to the silica shells this group of algae suffers from higher sinking losses compared with other phytoplankton. Thus, diatom blooms should result in enhanced sedimentation of N. Furthermore, nitrification, the crucial step prior to denitrification, may be enhanced in spring due to deeper penetration of oxygen into the sediment. Another phenomenon observed in these lakes is a depression in k_{TN} values in late summer (Fig. 5), which possibly can be attributed to nitrogen fixation by blue-green algae. As detailed species information are unavailable, this can hardly be proven by monitoring data. However, data show that the maximum of cyanobacteria biomass usually occurs in the relevant time. Taking into account the very low nitrogen to phosphorus ratios at many of the monitoring stations (TN/TP <5 from August–October, DIN/SRP <1 in August and September) the occurence of nitrogen fixing blue-green algae seems likely (see Windolf et al., 1996, for a discussion). As mentioned in the context of P retention, low settling rates could alternatively be responsible for a decrease in N retention.

In case of the Teltowkanal, the low seasonal variation in k_{TN} probably results from a steady supply of nitrate (about 80% of TN) by several large wastewater treatment plants accompanied by increased water temperatures in winter.

Beyond doubt, further investigation is needed to explain the spatial and temporal differences in N retention, e.g. the extraordinarily high values of k_{TN} in some of the lakes (Fig. 5). Shortcomings in the model boundary conditions as well as errors in the simulation of flow distribution must also be considered since both may result in misleading estimates of retention parameters. Furthermore, the approximation of TN retention as a first order process may turn out to be too simple when DIN/TN ratios vary largely.

A simple way to figure out nitrogen retention quantitatively is to compare cumulated N loads calculated by the calibrated TraM model with cumulated N loads from a conservative model run (Fig. 6). The latter simulation delivers nitrogen loads at a gage that would occur if N retention was negligible. As can be deduced from Fig. 6 about 30% of the total nitrogen which is discharged into the studied river section via its tributaries is retained. This results in a significant reduction of N export to downstream waters. Compared to estimated in-stream nitrogen retention in a Danish lowland river (Svendsen et al., 1998) nitrogen losses in the Havel River are remarkably high. Since denitrification is most effective at the sediment-water interface (Seitzinger, 1988), this might be explained to a substantial extend by the large surface area of the Havel Lakes (see also Behrendt and Opitz, 2000). Figure 6 illustrates that the deviation between model simulation and observation is rather small when looking at cumulated TN loads. However, as Table 3 indicates the temporal variability in TN concentrations is replicated by the model much less accurately.

3.3. Analysis of parameter sensitivity

As Figs. 3 and 5 indicate, the nitrogen retention parameter k_{TN} as well as the net phosphorus retention rate r_{TP} are subject to large spatial and interannual variation. The average parameter values used in model simulations are therefore associated with potentially high uncertainty. Thus, the sensitivity of simulation results to changes in parameter values was tested (Fig. 7). In case of k_{TN} the values found in calibration for each river section (Fig. 5) were varied by ±10, 25 and 50%, respectively. The values of r_{TP} were altered in the same range but only the months June–October with net P release (r_{TP}>0) were included.

As Fig. 7 shows, TN concentrations in summer are much more susceptible to errors in the retention parameter k_{TN} than winter concentrations. Multiplying k_{TN} by a scaling factor of 0.5 results in an increase in the monthly average TN concentration by about 36% in July but only 8% in February. Scaling factors of 1.1, 1.25 and 1.5 (retention enhanced by 10–50%) result in decreasing TN concentrations. Again the changes are more pronounced in the month with the lowest TN concentration (July) than in the month with highest C_{TN} (February). The outcome of the analysis can easily be interpreted: as expected, percental changes in the retention parameter show the largest effect when k_{TN} is high. Winter concentrations are less affected since nitrogen retention is low and C_{TN} is largely controlled by external loads. When looking at average TN concentations over the whole year the model seems robust since errors in k_{TN} must exceed 50% before C_{TN} changes by more than 20%.

For phosphorus (right graph in Fig. 7) results of the sensitivity analysis are again presented for the month with lowest (April) and highest (October) TP concentrations C_{TP}. Obviously, changes in phosphorus release rates in the period June–October do not have any effect on C_{TP} in April, indicating that P enriched lake water is completely flushed in winter at higher discharges. As expected C_{TP} shows a strong reaction to altered values of r_{TP} in October, since concentrations are heavily controlled by internal loading in late summer and autumn (see Sect. 3.1). Annual averages of C_{TP} are less susceptible to errors in r_{TP} estimates. Even a Δr_{TP} of ±50% results in average TP concentrations changed by not more than 15%.

3.4. The impact of reduced phosphorus emissions

Two selected scenarios of phosphorus control measures are presented here. In the first scenario ("P1") it was assumed that phosphorus emissions from point sources were diminished all over the Havel basin except for the subbasin of the tributary Spree, which was not in the focus of this study (Table 4). The reduction in P emissions from wastewater treatment plants was quantified by the use of the ArcEGMO-Urban model (Biegel et al., 2004) taking into account possible enhancements in P elimination. The second scenario ("P2") includes scenario P1 but comes with the additional assumption that the average TP load emitted from the Spree catchment is reduced by approximately 50% to reach a concentration level of about 80 μg l^{-1}. That is, the load reduction in the

Spree River was chosen to meet a certain target concentration (immission approach) as defined by a wastewater management plan (SSB, 2001). The base scenario with unchanged P emissions was called “P0”.

The relative effects of the pollution control measures are presented in Table 4. As indicated by the average emissions (g s^{-1}) for the base scenario P0, the Spree catchment alone contributes over 50% of the total P input to the Lower Havel River. Consequently, the reduction of P emissions from all other subcatchments by about 13% (scenario P1) has only little effect on the total P input (−5.2%). On the other hand, a 50% decrease in P loads of the Spree River and its side branch, the Teltowkanal, results in a significant change in total P input (−35.2%). The alteration of the average TP concentration of the Havel River turns out to be much lower than the change in average total emissions. This can directly be attributed to net phosphorus release from the lakes’ sediments, which (see Sect. 3.1) was assumed to be unaffected by altered external P loading.

Whereas Table 4 only shows long term average concentrations, the effect of reduced P emissions on the concentrations’ seasonality is pointed out in Fig. 8. The absolute decrease in mg l^{-1} is similar for all month. The relative rates of change (Δ TP (%)) however show substantial intraannual differences, since absolute concentrations also vary significantly due to internal loading and the natural flow regime of the Havel River. Whereas scenario P2 causes a reduction in TP concentration of up to 50% in spring the corresponding value for August–November is only 20% or less.

The consequences for water quality management are obvious: The TP concentration in late summer and autumn cannot be reduced to a level that results in phosphorus limitation of the phytoplankton, even if drastic pollution control measures would be implemented (see Sas, 1989, for a pragmatic definition of limiting concentrations). In contrast, concentrations in spring are more sensitive to decreased external phosphorus loading. As Fig. 8 illustrates, TP falls below 40 μg l^{-1} in April under scenario P2. Considering a TP/SRP ratio of about 0.3 in spring (data from lake E; see Fig. 1) dissolved phosphorus concentrations are dropped to a level where algal blooms might possibly be prevented by P limitation. As studies from the nearby Müggelsee (Spree catchment) by Köhler et al. (2000) suggest, initial nutrient concentrations control algal growth not only in spring but may also influence the succession of plankton over the whole year. Hence, a further reduction of external P load might be effective for eutrophication control in spite of persistent sediment P release.

3.5. The effect of reduced nitrogen emissions

As with phosphorus, different cases of reduced nitrogen emissions were analyzed with the TraM model but only a single scenario will be discussed here. In this scenario ("N1") it was assumed that non-point nitrogen emissions were reduced all over the Havel basin, again excluding the Spree subbasin. For this, landuse maps of present conditions (scenario "N0") were modified (Jacobs and Jessel, 2003), e.g. portions of arable land were converted to grassland and another portion was set aside. In addition, the growing of intercrops was considered in scenario N1 in order to minimize N losses. The change in N export from the catchment was calculated at the Potsdam Institute for Climate Impact Research using the SWIM model. Though TN loads from all considered subbasins were reduced by about 29% on average, the TN concentration of the Havel River decreases only slightly (Fig. 9). This emphasizes the outstanding impact of nitrogen loads emitted from the Spree catchment, which were not altered in scenario N1. That is, a significant reduction in average TN concentrations of the Lower Havel River would require massive emission control measures to be taken in the Spree catchment which comprises a number of large point sources. However, the effect of reduced N input on primary production is hard to predict: Summer DIN concentrations temporarily fall below 0.1 mg l^{-1} already under present conditions. It is uncertain whether a further decrease could induce prolonged N limitation or just favors N-fixing cyanobacteria as long as P concentrations are high. Another point in the discussion on reduced N emission are the costs, because it is expensive to upgrade N elimination in wastewater treatment whereas retention in the river system (see Sect. 3.2) is "for free".

4. Conclusions

By means of the simple nutrient transport model TraM it was shown that in-river turnover significantly influences total nutrient concentrations of the lowland river Havel. Hence, the quantification of nitrogen and phosphorus retention rates turned out to be essential not only for understanding the present state of water quality but also for evaluating possible strategies of river basin management. The TraM model, adapted to the characteristics of the Havel River, proved to be a useful and necessary extension to mesoscale catchment models. As TraM uses empirical rates to account for N and P retention, the applicability of the current model version is limited. Distinct changes in natural boundary conditions or management practice might render empirical coefficients examined by calibration invalid. Therefore, more complex models, which take into account all relevant species and reaction terms of the N and P cycle seem desirable. However, one must keep in mind the large number of parameters introduced by those models. Many of them are well defined in theory but hard to measure in the field and they are therefore prone to uncertainty. Effective settling velocities and the parameters involved in sediment phosphorus release are just two common examples, not to mention the problem of spatial heterogeneity. We think the approach taken in this study is a reasonable compromise, suitable for pointing out options and limits of nutrient concentration management. The next version of the TraM model will overcome some of its actual deficiencies. That is, a sediment compartment will be implemented to force closed mass balances (Reichert et al., 2001) and a generic reaction module is currently tested, which allows very simple to rather sophisticated kinetic models to be created by the user. In our opinion robust water quality models of adaptable complexity are needed to better integrate both, catchment nutrient dynamics and in-river/lake turnover into a modeling framework applicable in the context of the WFD.

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Table 1. Features of continuous flow stirred tank reactors (CFSTR) and plug-flow reactors (PFR) in the TraM model.

	CFSTR	PFR
Equivalent in nature	Polymictic lakes	River/channel sections
Advective transport	Neglected	Implemented
Dispersive transport	Implemented	Currently neglected
Transport calculation	Numerical solution of mass balance equation	Shifting of load hydrograph time axis according to flow velocity
Geometric input data	Stage-Volume- and Stage-Area-Relation	Flow area and wet perimeter as function of stage
Dynamic input data	Stage and flow hydrographs, time series of load from up-stream waters and external sources, retention parameters	

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Table 2. Monthly median values of TN retention (mg m^{-2} d^{-1}) for lakes of the Potsdamer Havel calculated from mass balances for the years 1996–2002. See Fig. 1 for lake labels C–F.

Lake	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
C	107	141	322	330	230	227	224	181	184	140	168	124
D	43	124	71	-15	36	42	133	44	27	46	16	-13
E	42	100	161	137	67	35	27	20	51	75	67	44
F	48	57	97	148	80	72	37	71	-8	90	66	20

Table 3. Errors in simulated TP and TN concentrations at monitoring station Ketzin (label 1 in Fig. 1) for the period of model calibration (1995–2000) and four additional years. ME: mean error (Bias), MAPE: mean absolute percental error, Efficiency: Nash/Sutcliffe index.

Variable	Parameter	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004
TP	ME (mg l^{-1})	−0.02	0.01	−0.02	−0.01	−0.02	−0.13	−0.01	0.02	0.10	0.05
	MAPE (%)	30.2	21.6	18.4	28.0	14.8	29.9	18.6	27.1	41.7	24.4
	Efficiency (–)	0.70	0.42	0.82	0.71	0.96	0.69	0.81	0.81	0.02	0.43
TN	ME (mg l^{-1})	0.18	−0.15	−0.33	0.07	0.30	0.08	0.17	0.02	0.12	0.05
	MAPE (%)	25.2	18.6	18.1	17.6	19.2	15.8	20.1	10.5	13.0	20.7
	Efficiency (–)	−0.19	0.10	0.03	−0.26	0.10	0.55	0.36	0.57	0.80	0.54

Table 4. Assumed reduction of TP emissions from the Spree catchment (E_{Spree}) and all other subcatchments (E_{Other}) and its effect on the average TP concentration of the Havel River C_{Havel} at monitoring station 2 (see Fig. 1). The initial values for the base scenario P0 are averages over 13 years.

Scenario	E_{Spree}	E_{Other}	E_{Total}	C_{Havel}
P0	6.2 g/s	4.2 g/s	10.4 g/s	0.23 mg/l
	ΔE_{Spree}	ΔE_{Other}	ΔE_{Total}	ΔC_{Havel}
P1	0.0%	−13.0%	−5.2%	−3.7%
P2	−50.0%	−13.0%	−35.2%	−23.8%

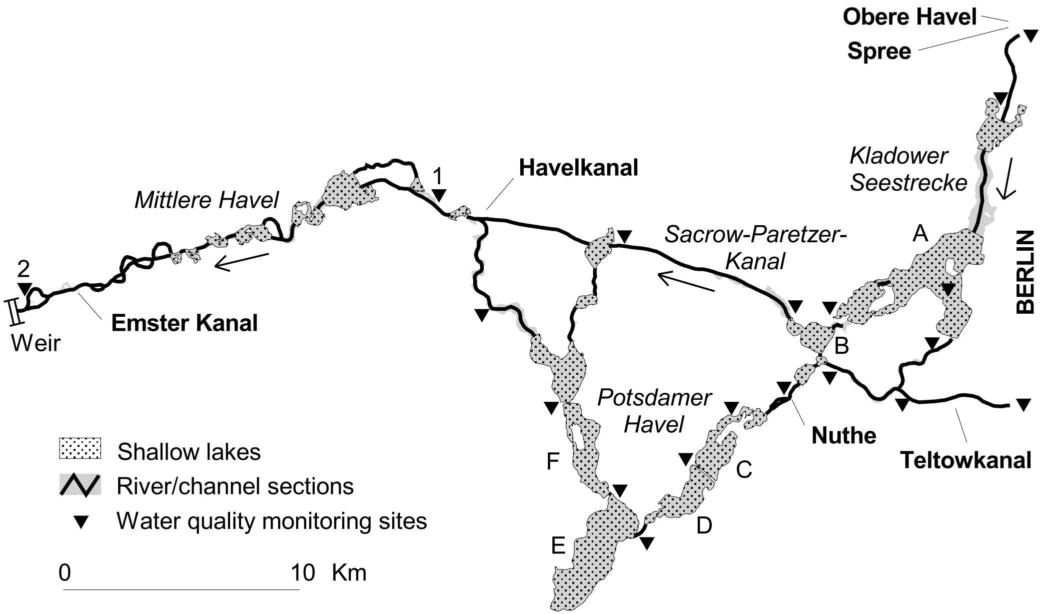

Fig. 1. The Havel River downstream of the city Berlin with its major inflows (in bold). Labels A–F and numbers are used throughout the text for reference purposes.

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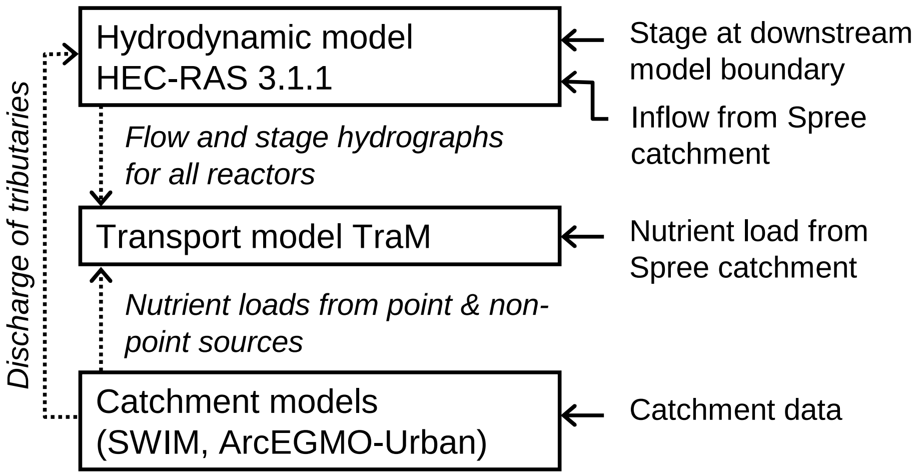

Fig. 2. Interaction between the transport, the hydrodynamic and the catchment model(s). Dotted arrows indicate the exchange of boundary conditions while solid arrows mark external input data.

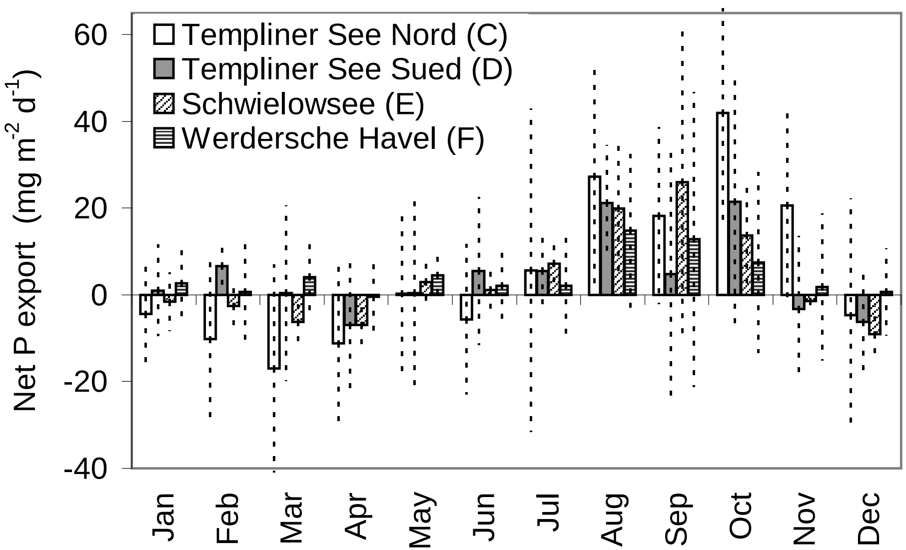

Fig. 3. Monthly medians of net phosphorus export (positive) and retention (negative) for 4 large lakes of the Lower Havel River over the period 1995–2000 (see Fig. 1 for the lakes' locations). The 95% confidence interval is indicated by dotted lines.

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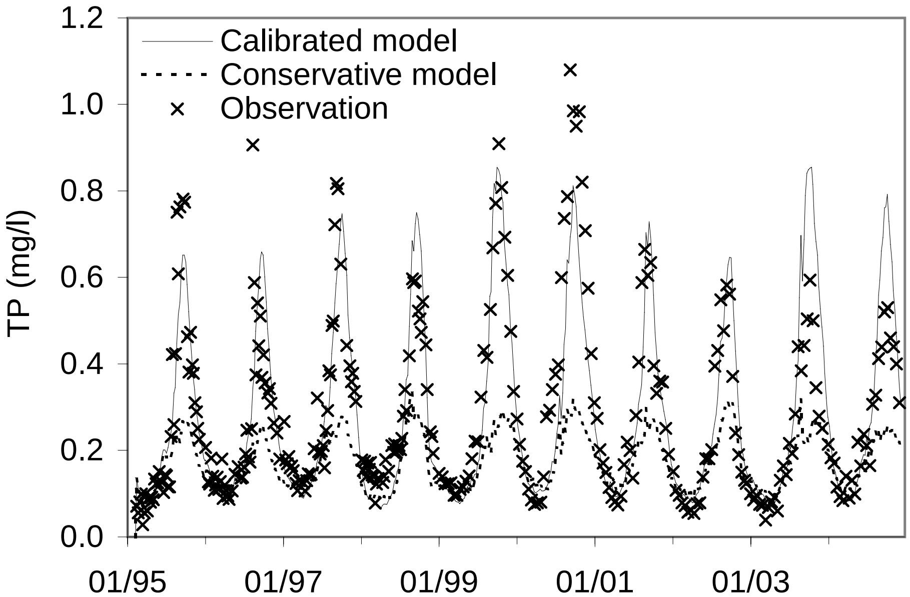

Fig. 4. Observed and simulated TP concentration of the Havel River at Ketzin (label 1 in Fig. 1) for the time period of model calibration (1995–2000) and four additional years.

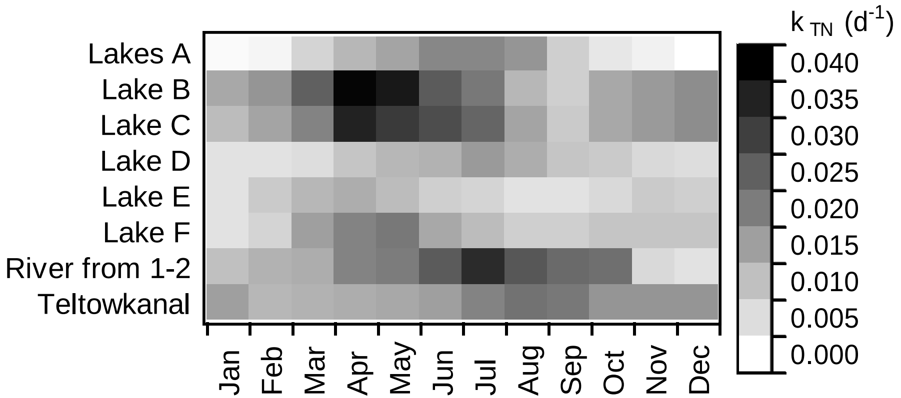

Fig. 5. Estimates of the rate constant k_{TN} (d^{-1}) used for approximating the retention of total nitrogen by a first order kinetics. The values were identified by model calibration in 1995–2000. Refer to Fig. 1 for labels.

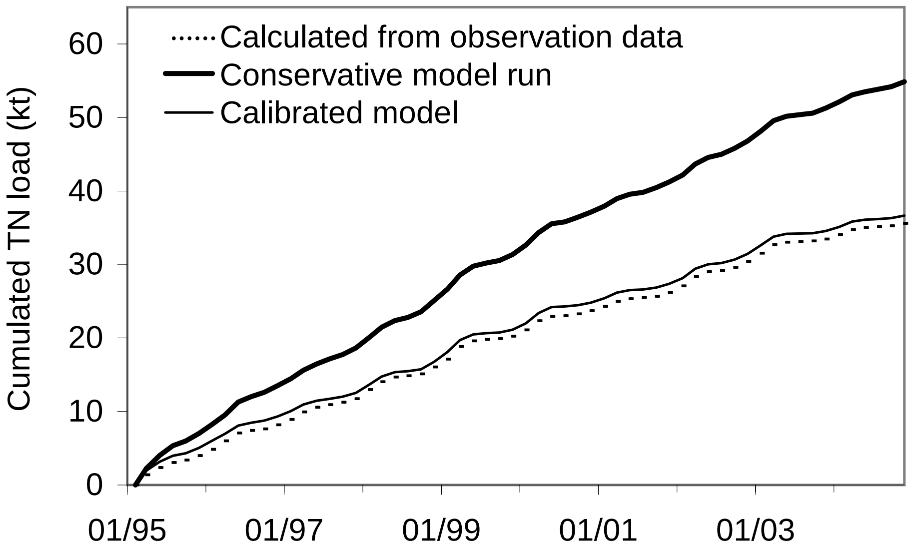

Fig. 6. Cumulated load of total nitrogen (kilotons) at the downstream end of the river network (monitoring station 2 in Fig. 1) as computed from observed data and model simulations.

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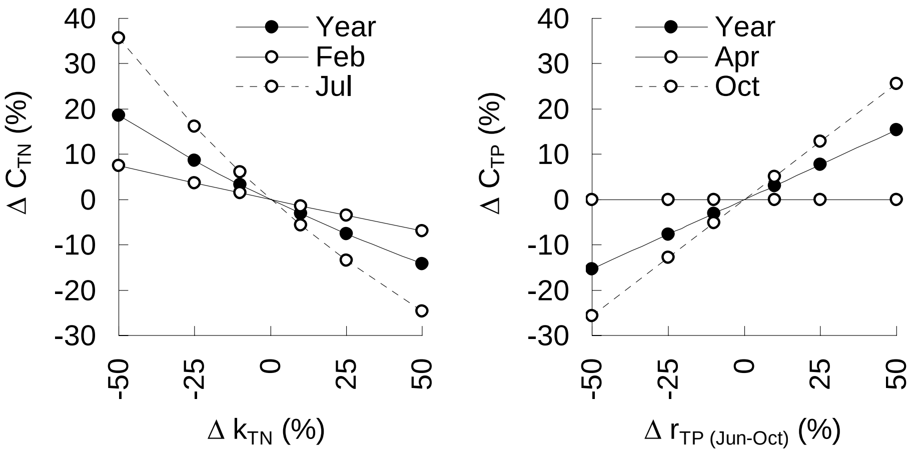

Fig. 7. Left: Simulated change in average TN concentrations ΔC_{TN} at monitoring station 2 (see Fig. 1) resulting from the use of altered retention parameters k_{TN} starting from calibrated values (Fig. 5). Right: Change in average TP concentrations ΔC_{TP} at the same monitoring station as simulated with net phosphorus retention rates r_{TP} for June–October varied from −50 to +50% around the mean values shown in Fig. 3.

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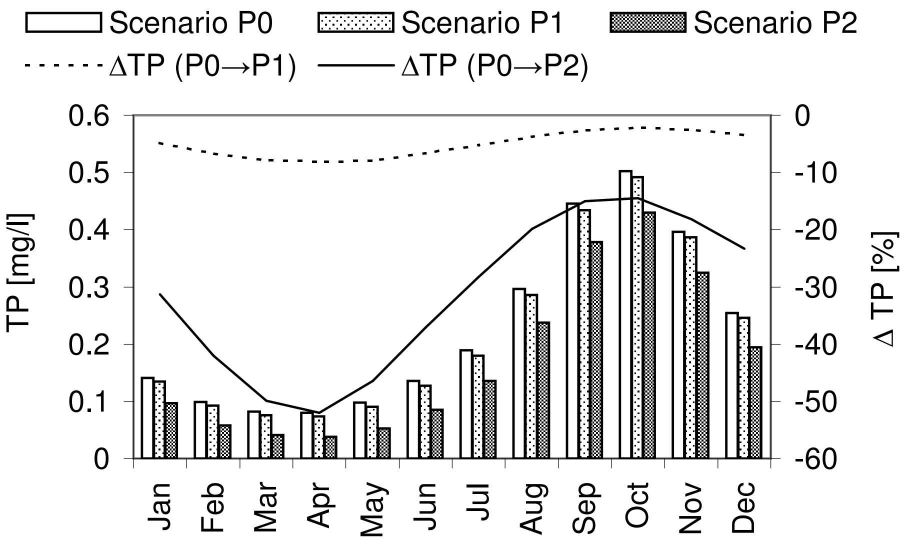

Fig. 8. Monthly averages of TP concentration (columns) at the downstream boundary of the studied river section (label 2 in Fig. 1) for the scenarios P0, P1 and P2. Lines illustrate the proportional change in average concentrations (Δ TP) associated with the scenarios P1 and P2.

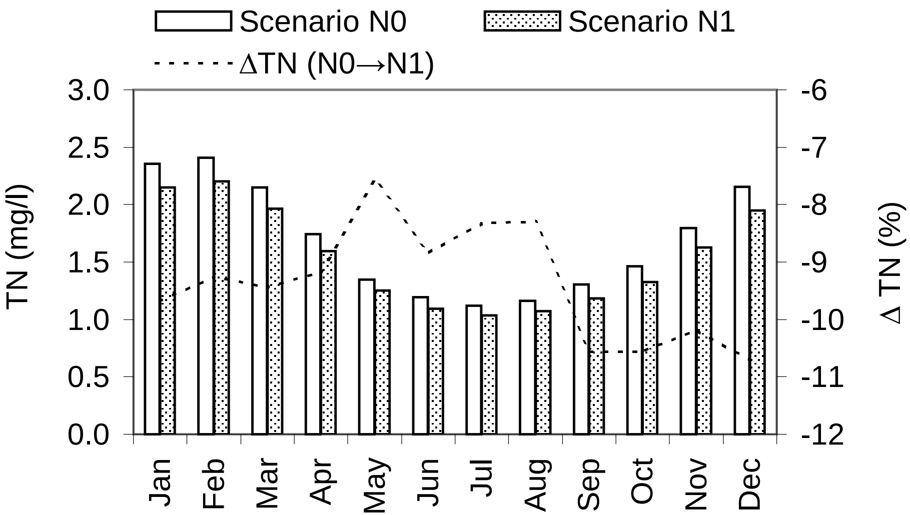

Fig. 9. Monthly averages of TN concentration (columns) at the downstream boundary of the studied river section (label 2 in Fig. 1) for scenario N0 and N1. The dashed line marks the proportional change in average concentration (Δ TN) associated with scenario N1.